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LCA OF WASTE MANAGEMENT SYSTEMS

Environmental impacts of end-of-life vehicles' management: recovery versus elimination

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Abstract
Purpose In Portugal, the management of end-of-life vehicles (ELV) is set out in targets of the European Union policy for the year 2015, including 85 % recycling, 95 % recovery, and maximum of 5 % landfilling. These goals will be attained only through more efficient technologies for waste separation and recycling of shredder residues or higher rates of dismantling components. Focusing on this last alternative, a field experiment was carried out. There is potential for additional recycling/recovery of 10 %.
Methods Three scenarios were proposed for the management of ELV wastes: (1) scenario 1 corresponds to the baseline and refers to the current management, i.e., the 10 % of ELV wastes are shredded whereby some ferrous and non-ferrous metals are recovered and the remaining fraction, called automotive shredder residues (ASR), is landfilled, (2) scenario 2 wherein the ASR fraction is incinerated with energy recovery, and (3) scenario 3 includes the additional dismantling of components for recycling and for energy recovery through solid recovered fuel, to be used as a fuel substitute in the cement industry. The environmental performance of these scenarios was quantified by using the life cycle assessment methodology. Five impact categories were assessed: abiotic resource depletion, climate change, photochemical oxidant creation, acidification, and eutrophication.
Results and discussion Compared to the other scenarios, in scenario 1 no benefits for the impact categories of climate change and eutrophication were observed. Scenario 2 has environmental credits due to the recycling of ferrous and non-ferrous metals and benefits from energy recovery. However, this scenario has a significant impact on climate change due to emissions from thermal oxidation of polymeric materials present in the ASR fraction. A net environmental performance upgrading seems to be ensured by scenario 3, mainly due to replacing fossil fuel by solid recovered fuel.
Conclusions The proposed additional dismantling of ELV (scenario 3) not only brings environmental benefits but also meets the European recovery and recycling targets. The associated increase of dismantling costs can be compensated by the additional recycling material revenues as well as social benefits by a rise in employment.

Keywords Automotive shredder residues (ASR) · End-of-life vehicles (ELV) · Life cycle assessment (LCA) · Management · Recovery operations · Solid recovered fuel (SRF)

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1 Introduction

The high production of vehicles and the rise of road traffic have had negative consequences. Negative environmental effects are however present not only during this phase but also at the end of it. At this stage, a vehicle gets the definition of end-of-life vehicle (ELV), i.e., it cannot fulfill the purpose for which it was produced.

Accordingly to the Eurostat statistics, every year motor vehicles which have reached the end of their useful lives generate around 6 million tons of waste in the European Union and should be managed correctly (Eurostat 2012).

On 21 October 2000, the European Union put into force the Directive 2000/53/EC—the "ELV Directive," in order to improve ELV treatment in an environmentally sound manner (EU 2000). This ELV directive covers aspects throughout the life cycle of a vehicle as well as related to treatment operations, such as: (1) reduction and control of hazardous substances in vehicles and the prohibition of the use of lead, mercury, cadmium, and hexavalent chromium; (2) dismantling of all end-of-life vehicles treated and recovered by the industry at no cost for its final holder in such a way that does not cause environmental pollution; and (3) the establishment of reuse, recycling, and recovery targets. With respect to the last issue, by 1 January 2015, the target for recovery and reuse is a minimum of 95 % of which a minimum of 85 % corresponds to reuse and recycling material. Consequently, less than 5 % (*w*/*w*) of the ELV would be landfilled.

The ELV treatment stages are: (1) pretreatment, a mandatory stage, which aims to remove most of the hazardous components such as batteries, fuels, lubricating oils, and refrigeration fluids; (2) dismantling, which consists of components removal from the car body in order to reuse them, if undamaged, or recycle material, such as glass from windscreens and plastic bumpers; and (3) shredding where the remainder of the ELV, so-called hulks, are turned into small pieces and the ferrous and non-ferrous metals are sorted by a series of mechanical and magnetic separation processes to be sent for recycling. The metallic fraction present in the shredded waste stream represents around 60 % of the total weight of the ELV (Ferrão and Amaral 2006; Nourreddine 2007). The unsorted waste stream derived from shredding corresponds to 15 % (*w*/*w*) of the original ELV weight and is collectively termed automotive shredder residue (ASR). The ASR is an agglomerate of nonmetal waste mainly plastics, rubber, textiles, fibrous materials, and wood, contaminated with metals and oil residues. Presently, in Portugal, 80 % of ASR is discarded as hazardous waste to landfill, when containing fluid and heavy metals, or by non-hazardous landfill depending on the results of chemical characterization. The remaining 20 % is incinerated with energy recovery (Valorcar 2011).

The treatment of ASR presents a major challenge for the European Union and also for Portugal. This is stimulating the development and application of new technologies and/or procedures for waste separation and recycling of shredder residues, such as post-shredder technologies (PST) and higher rates of dismantling components.

Due to the ASR heterogeneity in elemental composition and particle sizes, the economic and technical viability of its recycling and mechanical recovery is limited and complicated (Al-Salem et al. 2010; Fink 1999; Redin et al. 2001; Santini et al. 2010). Additionally, factors that prevent the total recovery of ASR include its contamination, physical nature, poor development of certain secondary markets, and high processing costs. Thermal treatment methods such as co-incineration or application as (energy) feedstock in cement industries may constitute an effective and a more sustainable cost alternative to ASR landfill. However, the ELV directive (EU 2000) implicitly establishes an upper limit of 10 % of ELV for energy recovery.

The stakeholders have secured the interest of some research groups to solve environmental problems associated with both ELV recovered components and discarding waste. The potential environmental impacts of each ELV management option can be quantified and subsequently considered in decision making.

The environmental management tool commonly used to perform that kind of studies is the life cycle assessment (LCA) methodology (Ciacci et al. 2010; GHK/Bios 2006; Jeong et al. 2007; Le Borgne and Feillard 2001; Puri et al. 2009; Schmidt et al. 2004; Sawyer-Beaulieu and Tam 2005, 2008).

This work is a contribution to the environmental assessment of different ELV management alternatives and includes a more extensive dismantling effort using ELV material recovery technologies. Different management scenarios are proposed and their environmental impacts are quantified.

2 Methods and procedures

Life cycle assessment is an environmental holistic tool for compilation and evaluation of the material and energy inputs and outputs of a given process in order to get a given product or service. From this inventory, it is possible to calculate the potential environmental impacts of the product throughout its life cycle. Thus, LCA is a tool for the analysis of the environmental burden of products at all stages in their life cycles—from the extraction of resources, through the production of materials, product parts, and the product itself, and the use of the product to the management after it will be discarded, either by reuse, recycling, or final disposal (i.e., "from the cradle to the grave"). This study has been conducted according to the EN ISO 14040:2006 and EN ISO 14044:2006 guidelines (ISO 2006a, 2006b).

2.1 Goal and scope definition

2.1.1 Goal of the study

Focusing on the strategy to increase the actual rate of dismantling of ELV in Portugal, a field experiment was carried out in a dismantling plant, accredited by Valorcar—Lyrsa Reciclagens Industriais, Unipessoal Lda.—in order to understand the current practices involved in the process. Considering the observed practices valid for all national dismantlers, additional materials and/or components with

recycling/recovery potential (currently not removed) were dismantled from different car brands. The dismantling of these materials is not mandatory by law and they have no commercial value because of their physical state or/and age. Their current route is the shredding stage, appearing afterwards in the ASR fraction.

The weight of additional dismantled materials corresponds to 10 % (*w*/*w*) of ELV and comprises: (1) wiring cables, (2) seats (composed of foams, plastics, textiles, and metals), (3) plastic components (headlights without market value, dashboard and other plastics easy to remove after the dismantling of other components), (4) textiles (seat belts, carpets, and linings), and (5) rubbers and interior sealants. The results classified by waste stream are reported in Table 1. The dismantling of these materials saves effort of separation in the shredding step and, ultimately, avoids their presence in the ASR fraction.

Focusing on that 10 % of ELV waste, three management scenarios are proposed and their environmental performance assessed. Table 2 summarizes the details of each scenario.

The first two scenarios are currently implemented in Europe for the ASR fraction, although landfill disposal is the most adopted management option across Europe (Eurostat 2012) and Portugal is not an exception.

Ladeira (2002) gives average separation efficiency upon shredding of 96, 39, and 60 % for ferrous metals, copper, and aluminum, respectively. These values were utilized in the present work (scenarios 1 and 2).

Scenario 3 is proposed as a case study for supplementary dismantling, which represents a management strategy, via processes to recover both material and energy. More information and details about all scenarios are reported in Section 2.3.

The methodology of LCA was applied in order to assess the environmental performance of the three ELV management scenarios, using the EcoInvent (2010) and the ELCD (2010) databases and CML 2001 procedure.

Table 1 Waste flows arising from the additional dismantling of ELV

Waste flow	Composition	Mass [kg/ELV[a]]	Average percentage, *w/w* [%]
Ferrous metals	Steel	19.5	2.0
Non-ferrous metals	Aluminum	1.5	0.2
Plastics (mixture)	Plastics mixture	23.0	2.3
Wiring cables	Plastic	7.0	0.7
	Copper	4.1	0.4
Foams	Polyurethane	13.5	1.4
Textile fibers	Including nylon and other tissues	24.9	2.5
Rubbers	Rubbers	6.5	0.7
	Total	≅ 100	10

[a] Considering an average European ELV weight of 1 ton (Zoboli et al. 2000)

2.1.2 Scope of the study

Functional unit The functional unit (FU) adopted in present work is 10 % of one average ELV, i.e., 100 kg of ELV composed of different materials (plastics, ferrous and non-ferrous metals, textiles, foams, and rubbers) which are currently not removed by Portuguese dismantlers (see Table 1).

System boundaries The system boundaries must be identified clearly in order to establish included processes as well as those that could be discarded at a given cutoff criterion.

According to ISO 14040 and 14044 (ISO 2006a, 2006b), a cutoff criterion was applied to all flows occurring in life cycle stages prior to dismantling (including collection and transportation to dismantling), which is consistent with the principle of excluding equivalent activities for LCA. However, scenarios 1 and 2 do not include the dismantling process.

System boundaries begin with the geographical and physical boundaries of each waste management scenario and they close with the landfill disposal or with the benefits resulting from energy and material recovery. In addition, environmental flows and burdens related to each waste treatment plant and environment were considered. The study focuses on the Portuguese context (electricity production and consumption, transport, etc.).

2.2 Life cycle analysis methodology

After goals and scope definition, the methodology of LCA must outline each scenario and the processes to be considered within it. Then, material and energy flows must be established for each process. This can include technological and material resources and the producer's technical resources and emissions to the environment. To evaluate the overall environmental impacts of each scenario, a life cycle inventory analysis (LCI) is performed and the life cycle impact assessment (LCIA) is established. The LCIA calculation used in this work is supported by Excel® spreadsheets.

2.2.1 Information sources and data quality requirements

The quality of a LCA study is largely dependent on the quality of the information upon which it is based.

In this study, on-site data were used whenever possible. For the dismantling stage in particular, site-specific data were measured and used. Those data represent the usual type of dismantling practice/technology used in Portugal. The remaining data were obtained in specific and technical literature or, if unavailable, they were estimated.

Table 2 Management scenarios proposed in this study

Scenarios	Description
Scenario 1—landfilling	Baseline scenario. Current ELV waste fraction treatment. The components are not dismantled but shredded. The ASR fraction is landfilled.
Scenario 2—thermal treatment with energy recovery	The ASR fraction is co-incinerated with municipal solid wastes (MSW) with energy recovery.
Scenario 3—supplementary dismantling	Supplementary dismantling of ELV is considered, with additional separated streams of recyclables and production of SRF, for further use as fuel in a cement industry.

Additionally, the EcoInvent Database v2.2 (2010) by Swiss Centre for Life Cycle Inventories and the ELCD (2010) were used as references to the processes of energy production, transport, shredding process, infrastructure, and recovery or disposal of waste in each scenario modeled.

2.2.2 Assumptions of the study

The data and information that describe sources of pollutants in the ASR fraction, such as oils and fuels, were not incorporated into the FU definition. This limitation of the study may be justified by assuming that all pollutants were ideally removed during drainage operations according to the European Directive requirements. Furthermore, the study of Morselli et al. (2010) shows that ASR chemical–physical parameters remain under the hazard threshold.

2.2.3 Life cycle inventory analysis

In this study, LCI analysis has been carried out in accordance with the procedure described in ISO 14040 and 14044 guidelines (ISO 2006a, 2006b).

Inventories were prepared for material and energy flows, inputs, and environmental emissions for the three scenarios. The LCI processes of each scenario are from EcoInvent Database v2.2 (2010) and ELCD (2010). These inventories include all natural resources used in the production of technological resources needed to manufacture the given goods and, naturally, all emissions to air, water, and soil.

2.2.4 Life cycle impact assessment

According to ISO 14040 and 14044 (ISO 2006a, 2006b), CML 2001 method was adopted to characterize and quantify the main impacts in terms of potential effects on the following five environmental categories: (1) abiotic resource depletion (ARD), (2) climate change (CC), (3) photochemical oxidant creation (POC), (4) acidification (AC), and (5) eutrophication (EU). More details are provided by Table 3 which describes the characterization factors and models for determining the values of the previous impact categories.

The classification and characterization steps were carried out to evaluate the impact profile of each treatment system considered, while facultative steps such as normalization and weighting were excluded to minimize the subjective elements of the LCIA study.

In this study, the environmental burdens of ELV dismantling and recycling processes are calculated and the environmental benefits resulting from the avoidable effects of recycled materials are quantified. Thus, the total environmental impacts are computed by deducing the benefits to the environmental burdens.

2.3 Scenario development

Material and energy flows inputs and environmental emissions were quantified in each scenario. The main details and information are described below.

2.3.1 Scenario 1—landfilling

Scenario 1 represents the current management of the FU (100 kg/ELV), i.e., the components are not dismantled and are processed in a shredding plant, where ferrous scrap and non-ferrous metals are recovered by eddy current machines combined with an innovative induction sorting system. The ASR fraction generated in this process is landfilled. The transport of ELV to shredding as well as of ASR to landfill and the infrastructure facilities were included. Data on the infrastructure (e.g., mechanical treatment plant, landfill facility, wastewater treatment plant, sewer grid, incineration plant, slag compartment) include: (1) construction of infrastructure (materials, construction processes and facilities such as machinery, cables, etc.) and (2) disposal activities (dismantling processes and disposal infrastructures) taking into account the processing capacity, annual production, and the facility's lifetime. In cases where transport takes place, construction, road use, operation, and dismantling per unit are also included.

The scheme of this scenario is shown in Fig. 1. Considering the FU composition (see Table 1) and the separation efficiency after shredding (Section 2.1.1),

Table 3 Environmental impact categories and their indicators, characterization factors, and characterization models considered in this study (Adapted from Heijungs et al. 1992)

Impact category	Indicator	Characterization factor	Characterization model
Abiotic resource depletion (ARD)	kg Sb eq	Abiotic depletion potential (ADP)	Based on ultimate reserves and extraction rates
Climate change (CC)	kg CO_2 eq	Climate change potential for a 100-year time horizon (CCP_{100})	Calculated based on the model developed by the Intergovernmental Panel on Climate Change (IPCC 2006) defining the global warming potential of different greenhouse gases
Photochemical oxidant creation (POC)	kg C_2H_4 eq	Photochemical ozone creation potential (POCP)	Calculated based on the UNECE Trajectory model, which indicates the potential capacity of volatile organic compounds (VOCs) to produce ozone
Acidification (AC)	kg SO_2 eq	Acidification potential (AP)	Calculated with RAINS10 model, developed at IIASA, describing the fate and deposition of acidifying substances, adapted to LCA
Eutrophication (EU)	kg PO_4^{3-} eq	Eutrophication potential (EP)	Based on stoichiometric procedure, which identifies the equivalence between N and P for both terrestrial and aquatic systems

scenario 1 allows a separation of 18.7 kg of ferrous metals (steel) and 2.5 kg of non-ferrous metals (copper and aluminum) from 100 kg ELV. The remaining fraction composed of 30.0 kg of plastics, 6.5 kg of rubbers, 13.5 kg of foams (polyurethane), 24.9 kg of textile fibers, and 3.9 kg of metal fragments, amounting to 78.8 kg, is landfilled.

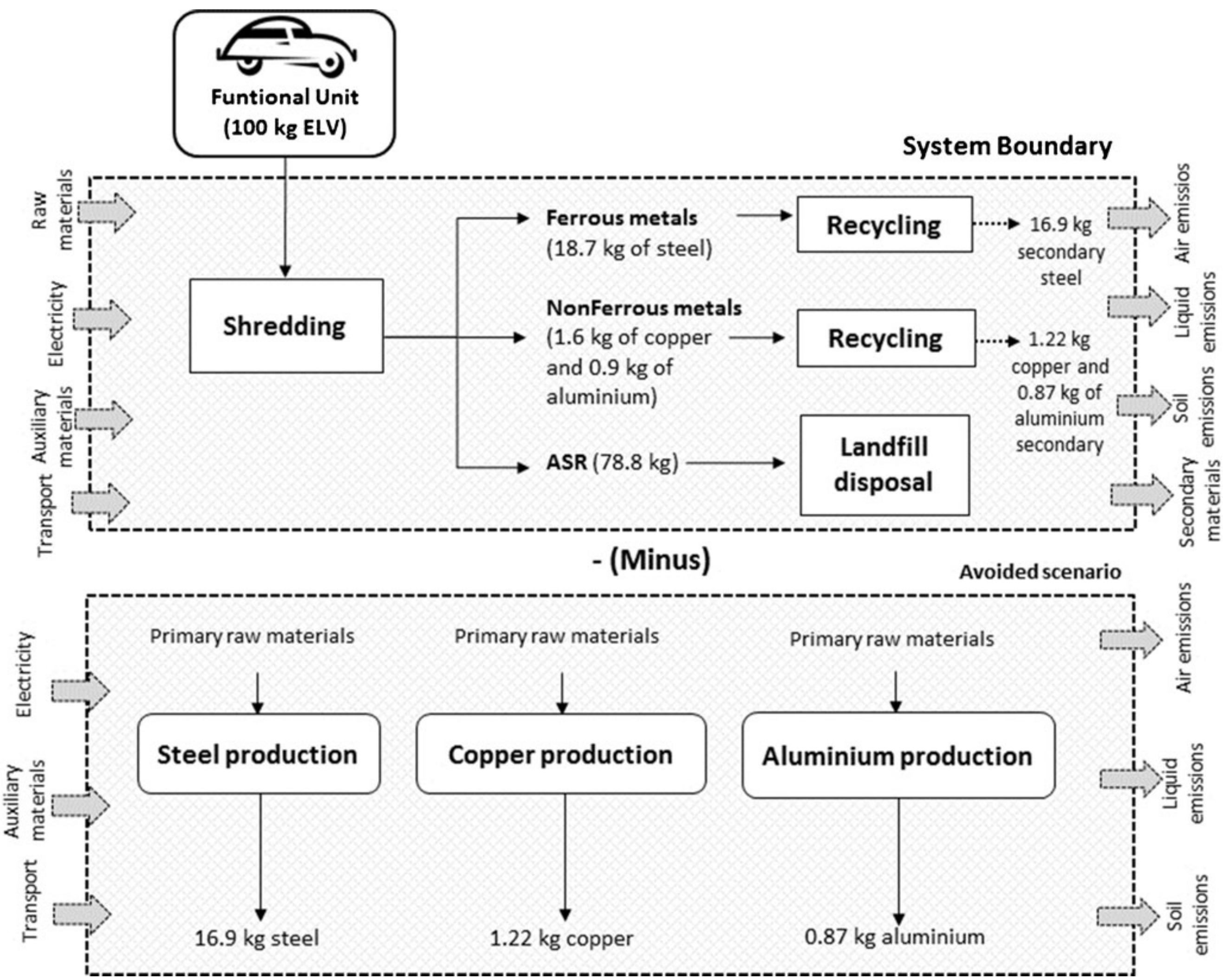


Fig. 1 Boundaries of scenario 1

Concerning the recycling processes to produce steel, copper, and aluminum, it should be noted that 1 kg of each material scrap does not give rise 1 kg of "secondary materials," but 0.9, 0.76, and 0.97 kg of each metal, respectively (Classen et al. 2009).

Modeling waste landfilling involves a lot of critical aspects due to the difficulty to establish unambiguous relationships among wastes and their environmental impacts. Indeed, these impacts will depend on the waste composition. In the databases consulted for this study, a process was not found that represented the landfilling of ASR. Given this lack of information, it was necessary to build a global process for the ASR fraction disposal. For that, the Ecoinvent (2010) database was used as flow reference related to the different processes of disposal of alternative materials, appropriately weighted, such as non-ferrous metals disposal (process: *disposal*, *aluminum*, *0 % water*, *to sanitary landfill*), ferrous metals disposal (process: *disposal*, *steel*, *0% water*, *to inert material landfill*), mixture of plastics and rubbers disposal (process: *disposal*, *plastics*, *mixture*, *15.3% water*, *to sanitary landfill*), and PUR foams disposal (process: *disposal*, *polyurethane*, *0.2% water*, *to sanitary landfill*). In addition, the ELCD (2010) database was used for textile fibers disposal (process: *landfill of textiles*).

In landfill process, the leachate is collected during the first 100 years and treated in a municipal wastewater treatment plant. The resulting wastewater sludge is incinerated in a municipal incineration plant. The incineration residues are landfilled in slag compartments and residual material is disposed off in a landfill.

The waste landfilling leads to direct air and water emissions as well as burdens on land use. Indirect burdens originate from the consumption of energy and infrastructure materials. The inventory data of this process contain all the burdens from the landfill, from the wastewater treatment, and from incineration slag/residual landfills.

2.3.2 Scenario 2—thermal treatment with energy recovery

The difference between scenarios 1 and 2 lies in the fate of the ASR fraction: in scenario 2 (Fig. 2), it is co-incinerated with MSW with energy recovery, whereas in scenario 1, it is landfilled. It is assumed that the inputs and outputs (emissions) of ASR/MSW co-incineration are similar to those of MSW incineration. This assumption is corroborated by the study of Ciacci et al. (2010), in a North Italian medium-sized incineration plant, where no changes in inputs and outputs were observed in co-incinerating 5 % ASR with MSW. The co-incineration of these wastes is frequent in European countries and the rates range between 3 and 11 % (Eurostat 2012).

According to Roy and Chaala (2001), the ASR has an average calorific value around 20 MJ/kg (16.9–30.7 MJ/kg). Assuming this value in scenario 2, in the co-incineration of 78.8 kg ASR/FU, it is possible to recover 55.7 kWh/FU of electrical energy and 407.7 MJ/FU of thermal energy. This thermal process produces 6.3 kg of slag and 0.9 kg/FU of fly ashes and scrubber sludge solids, which corresponds to 8 and 1.1 % of the whole input, respectively. These output wastes are landfilled after being made inert. Other needs for auxiliary processes, such as waste transport to a shredding plant, to the incinerator and infrastructure facilities were also included in the inventory.

To account for the environmental benefits obtained from the electrical energy and heat produced, these were compared with production by using virgin raw materials.

For the LCI of electricity production, the following energy mix substituted from nonrenewable sources was considered: 52.3 % hard coal, 35.0 % natural gas, 1.8 % fuel, and 10.9 % nuclear. This mix of energy sources is the currently used in Portugal (EDP 2011).

Since no data for Portuguese heat production was found, the following energy mix was adopted for steam production in a modern German waste incinerator: 34.52 % hard coal, 47.98 % natural gas, 11.35 % brown coal, and 6.15 % fuel oil (GHK/Bios 2006).

2.3.3 Scenario 3—supplementary dismantling

The scenario 3 (Fig. 3) arose from the field work experiment in a dismantling plant. As stated in Section 2.1.1, it was realized that there is a potential to increase by 10 % (*w*/*w*) the recycling/recovery rates of the ELV, by dismantling the additional components (see Table 1). Thus, in this scenario, those components are dismantled in a sequence of operations that essentially involve human labor and electricity consumption assumed to be 4 kWh/FU. The transport of materials inside the dismantling unit was neglected. The dismantled wiring cables amounting to 11.1 kg/FU are sent for physical and mechanical treatment in order to recover 7.0 kg/FU of copper with status of a secondary raw material and 4.1 kg/FU of wire plastic which is sent for MSW incineration with energy recovery.

The remaining dismantled components, amounting to 88.9 kg/FU, are composed of a wide range of materials. They were considered non-hazardous industrial waste, with no market value, but can be profitably converted into solid recovered fuel (SRF). This fuel provides an alternative to more expensive fuels such as coal, coke, oil, natural gas, and other fossil fuels, and is compatible with all major combustion systems, including the cement, paper, or metal industries.

The materials of this fraction are processed so that ferrous metal (from seats) and non-ferrous metals (from dashboard) are recovered and recycled, and the light fraction corresponds to SRF to be further used as fuel in the cement industry. The classification system for SRF (CEN 2006; NP 2008), is based on the limit values for the following

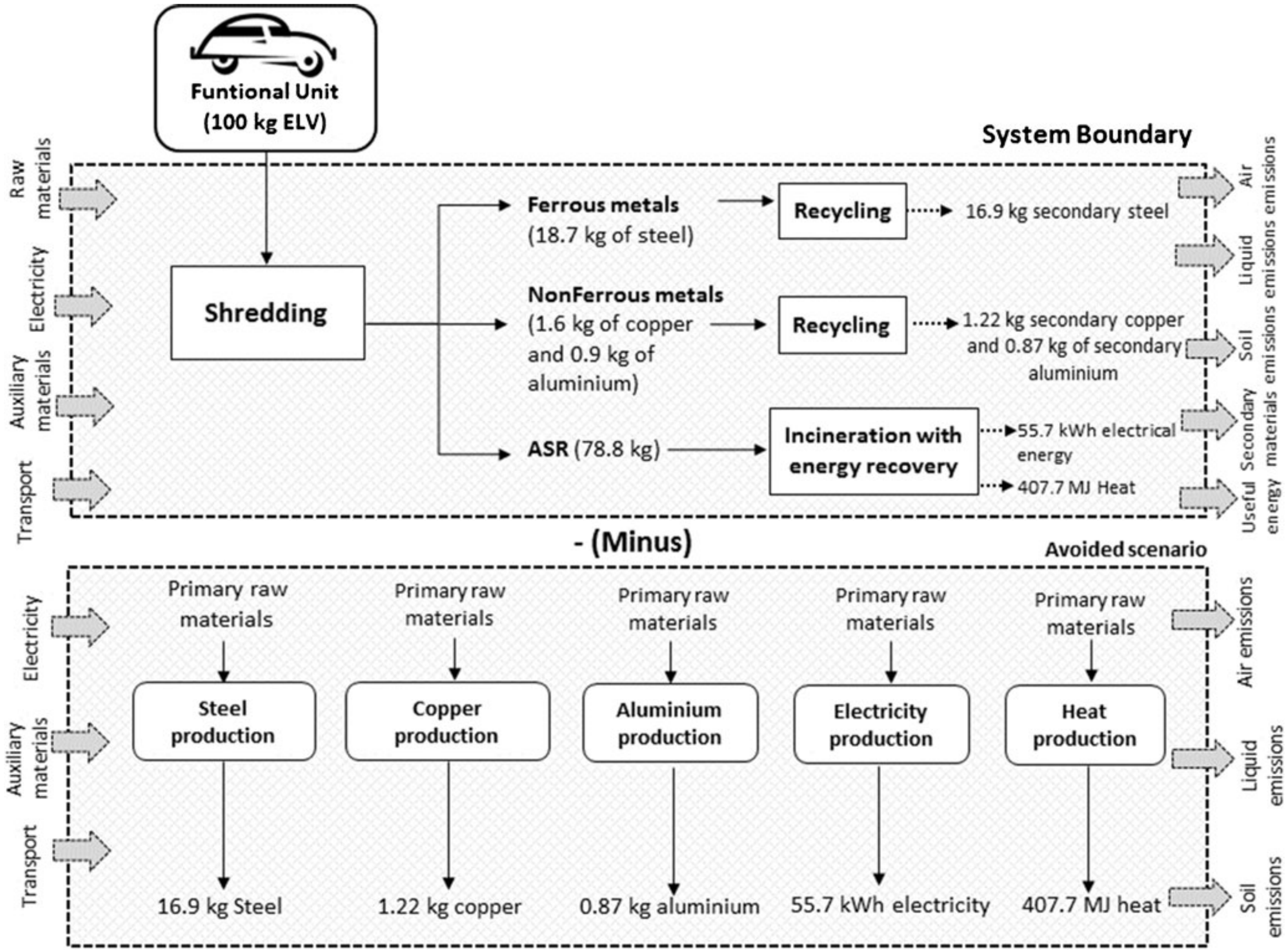


Fig. 2 Boundaries of scenario 2

three parameters: (1) an economic parameter (net calorific value (NCV)), (2) a technical parameter (chlorine content), and (3) an environmental parameter (mercury content). In this work, it is assumed that SRF produced in scenario 3 belongs to class 2, i.e., it has NCV ≥20 MJ/kg air mean, chlorine content ≤0.6 % dry, and mercury content ≤0.06 mg/MJ air.

In short, this scenario allows a separation of 25.1 kg/FU of ferrous and non-ferrous metals, which are sent for recycling and 67.9 kg SRF/FU that is burnt in the cement industry.

In the original process, to produce 1 kg of cement, an input of the following fuels is required: 0.0354 kg of coal, 0.0255 of heavy fuel, 0.000374 kg of light fuel oil, and 0.00391 kg of petroleum (Kellenberger et al. 2007) equivalent to a total amount of 2.14 MJ/kg cement. In agreement with Dias (2011), in the European market, high-quality SRF replacement ranges between 15 and 30 % in cement kilns. In this work, it was assumed that 20 % (0.43 MJ/kg cement) of the energy requirements in the cement kiln is covered by SRF (with 20 MJ/kg SRF). Since 67.9 kg of SRF/FU is generated in scenario 3, it enables a 1,358-MJ/FU of fuel replacement in the cement industry. Similar emissions from SRF incineration and burning fossil fuels were considered in that industry.

In this scenario, the environmental burdens of the several processes were quantified as well as the avoided impacts from material and energy recovery. The replacement of fossil fuels by SRF was also quantified and subtracted.

3 Results and discussion

Table 4 summarizes the information for the three scenarios, including the amounts of material and energy recovered and the landfilled waste.

In terms of material recovery, scenario 3 contrasts with the other two by the additional 3.9 kg/FU achieved.

The energy recovery only happens in scenarios 2 and 3. Scenario 2 has an electrical energy recovery ten times higher than scenario 3, due to the high amount of incinerated waste. On the other hand, the thermal energy recovery in scenario 3 is 3.4 times greater than in scenario 2.

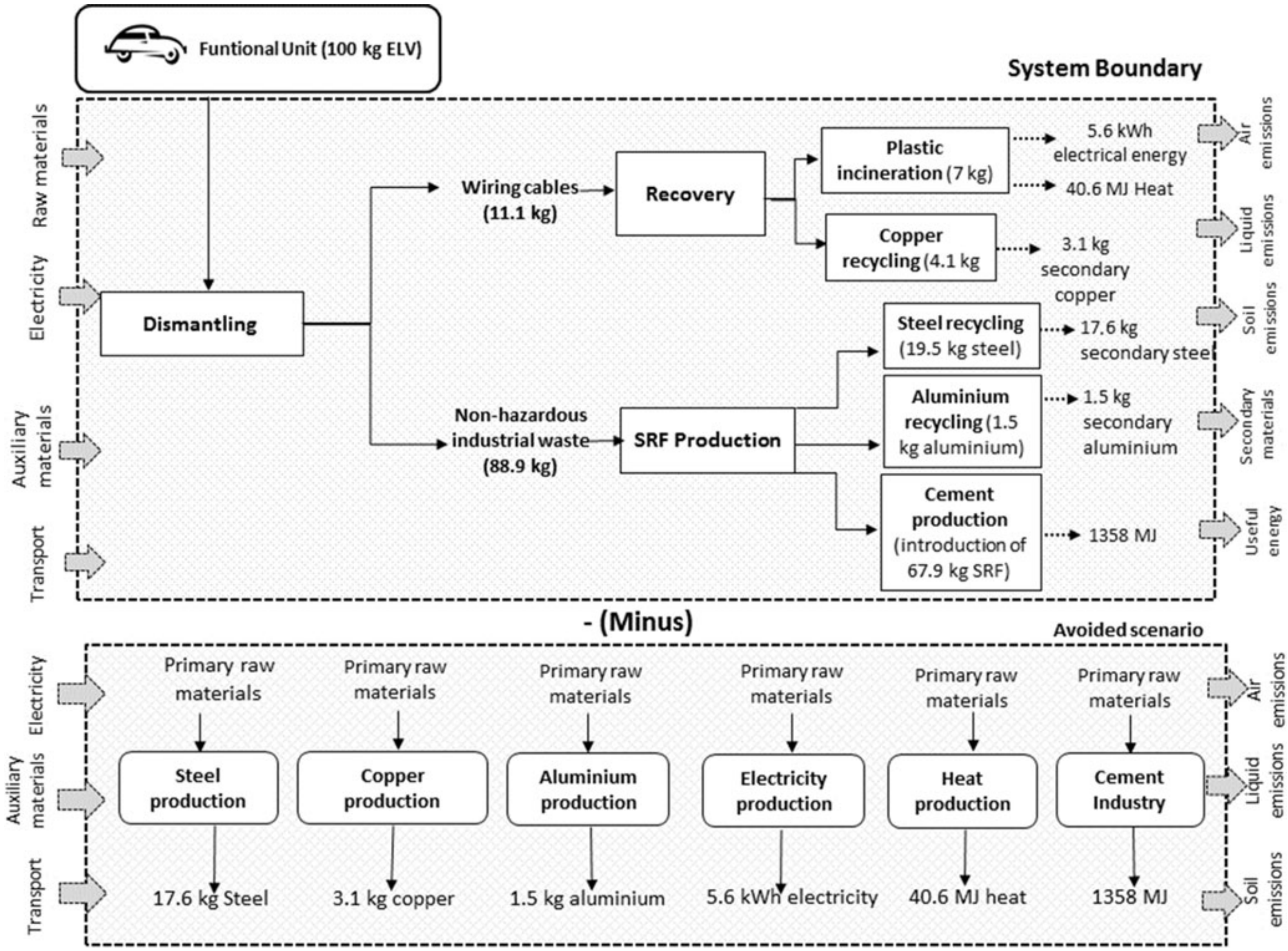


Fig. 3 Boundaries of scenario 3

With respect to the quantities of waste disposed in landfill, significant differences are observed among the three scenarios. Scenario 1 exhibits the higher disposal rate, i.e., around 11 and 143 times greater than scenarios 2 and 3, respectively. In scenario 2, the waste disposal is 13 times that of scenario 3.

The total environmental impact of each scenario is shown in Fig. 4.

Although some material recovery is observed in scenario 1, it has the worst environmental performance of the three. This is a consequence of the ASR landfill, which represents a net loss of material. The results show no benefit for the impact categories of CC and EU in this baseline scenario, exhibiting positive values of 4.9 kg CO_2 eq/FU and 0.27 kg PO_4^{3-} eq/FU, respectively.

In scenario 2, the co-incineration of ASR shifts polymeric materials away from the landfill. This would allow decreased damage associated with disposal and further benefits from energy recovery, such as destruction of organic pollutants and reduction in wastes volume. However, comparing this scenario with the remaining one, it has a significant impact on the climate change potential (20 kg CO_2 eq/FU) associated with emissions from oxidation of polymeric materials present in the ASR fraction. However, it has the better environmental performance in POC and AC categories.

Only scenario 3 exhibits negative values (−41 kg CO_2 eq/FU) on climate change category. This environmental benefit can be gained by replacing fossil fuels by SRF in the cement industry. This process contributes to the conservation of nonrenewable resources, avoiding the burning of fossil fuels and emission of CH_4 and CO_2. Scenario 3 also has the largest benefit for ARD category (1.2 kg Sb eq/FU) due to virgin raw materials being replaced by SRF and recycled materials.

4 Sensitivity analysis

According to ISO 14040 (2006a), the interpretation stage of an LCA study provides a sensitivity analysis. This kind of procedure is used to evaluate the effects of the assumptions and estimates of data used in the study.

Table 4 Main outputs (per FU) considered in each scenario

Scenario	Material recovery	Energy recovery	Landfilling
1	21.2 kg of metals	–	78.8 kg of ASR
2	21.2 kg of metals	55.7 kWh of grid electricity and 407.7 MJ of thermal energy	6.3 kg of slag, 0.9 kg of fly ashes and scrubber sludge solids
3	25.1 kg of metals	5.6 kWh of grid electricity and 1,398.6 MJ of thermal energy	0.43 kg of slag, 0.12 kg of fly ashes and scrubber sludge solids

In this work, a sensitivity analysis was performed for some inventory parameters, namely (1) separation efficiency of shredding process (scenarios 1 and 2) and (2) fossil fuel replacement by SRF in cement kiln (scenario 3). The reference scenarios, which result were presented and discussed in the previous section, are taken as "reference" and so-called a. For scenarios 1 and 2, separation efficiencies were changed ±10 %, originating in scenarios "b" and "c." For scenario 3, the SRF replacement percentage was varied ±10 % giving rise to two scenarios with the same designations "b" and "c" created for both scenarios 1 and 2. Table 5 summarizes the data used in this sensitivity analysis.

The results are shown in Fig. 5 for the five impact categories used in the present work.

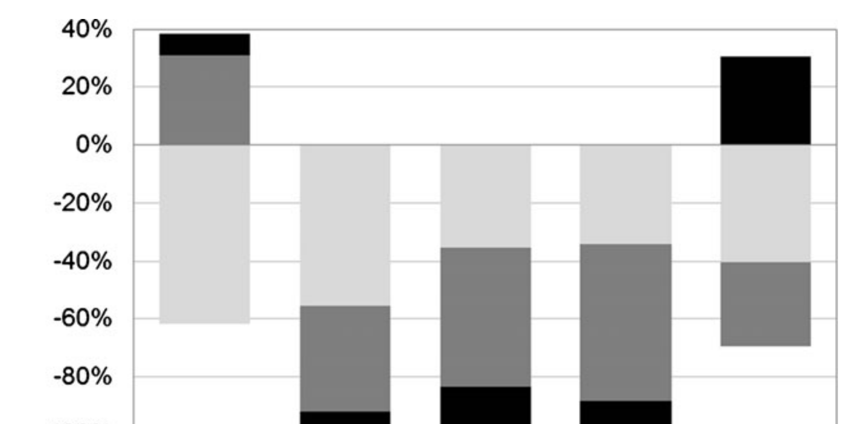
In scenario 1, the variation of 10 % in the shredding process efficiency has a remarkable influence on the climate change, with a decrease of 47 % for scenario 1b and an increase of 78 % for scenario 1c. The eutrophication is the less affected impact category presenting a change around 12 %. It can be concluded that increasing the separation efficiency during shredding is highly significant in reducing all evaluated environmental impacts. Analogous behavior is observed in scenario 2, but with lower variations, i.e., 10 % variation in efficiency does not influence significantly the LCA results. Climate change has the higher change (an increase of 19 % for scenario 2c) and abiotic resource depletion has the lowest (a decrease of 2 % for scenario 2b).

Finally, the variation of 10 % in the fossil fuel replacement by SRF has a significant influence in two environmental impact categories of scenario 3: abiotic resource depletion (38 %) and climate change (25 %). The eutrophication has the lowest change (5 %). An increase of replacement percentage leads to a reduction of all impact categories, due to the saving of natural resources.

5 Recycling and recovery rates

Beyond the environmental impact assessment of the three management scenarios developed in this work, it is interesting to consider the recycling and recovery rates attainable and compare them with the European Union targets for 2015. Table 6 compiles these values.

The recycling and recovery rates of 84.2 and 88.3 %, respectively, of scenario 1 correspond to the Portuguese average percentages registered on the Valorcar website (2011). The recovered materials of the FU are included in the current recycling rate and the remaining fraction in the landfill rate (11.7 %).



	CC	ARD	POC	AC	EU
scenario 1	4.9E+00	-1.7E-01	-1.4E-02	-1.0E-01	2.7E-01
scenario 2	2.0E+01	-7.9E-01	-4.2E-02	-4.9E-01	-2.5E-01
scenario 3	-4.1E+01	-1.2E+00	-3.1E-02	-3.1E-01	-3.5E-01

Fig. 4 Environmental performance of ELV management scenarios

Table 5 Data inventory parameters for sensitivity analysis

	Scenario 1 (separation efficiencies)	Scenario 2 (separation efficiencies)	Scenario 3 (SRF replacement)
a	Ferrous metals—96 % Copper—39 % Aluminum—60 %	Ferrous metals—96 % Copper—39 % Aluminum—60 %	SRF—20 %
b	Ferrous metals—100 % Copper—49 % Aluminum—70 %	Ferrous metals—100 % Copper—49 % Aluminum—70 %	SRF—30 %
c	Ferrous metals—86 % Copper—29 % Aluminum—50 %	Ferrous metals—86 % Copper—29 % Aluminum—50 %	SRF—10 %

None of the scenarios fulfill the recycling rate stated for 2015 by the Directive 2000/53/EC. Nevertheless, scenario 3 comes close. In terms of recovery rates, both scenarios 2 and 3 reach and even exceed the established target.

Furthermore, this analysis shows that an extended dismantling of vehicles (scenario 3) has the best environmental performance and meets the European recovery and recycling targets for ELV.

6 Conclusions

LCA methodology was used in this study in order to evaluate the environmental performance resulting from three different management scenarios for dealing with ELV waste.

According to the results, measures aimed at materials recovery not only meet the European recycling and recovery targets for ELV but also give greater environmental benefits compared to current practices. Three scenarios were considered:

1. Scenario 1 (baseline) is worst due to the environmental impact from the disposal of hazardous waste such as ASR, resulting in a net loss of material. No benefit for the impact categories of climate change and eutrophication potential is observed in comparison with the other two scenarios.
2. Scenario 2 allows decreased damage associated with landfilling polymeric materials and further benefits from energy recovery. Moreover, thermal treatment of ASR considerably reduces its volume and mass: only the resulting inert ash must be landfilled. Despite the

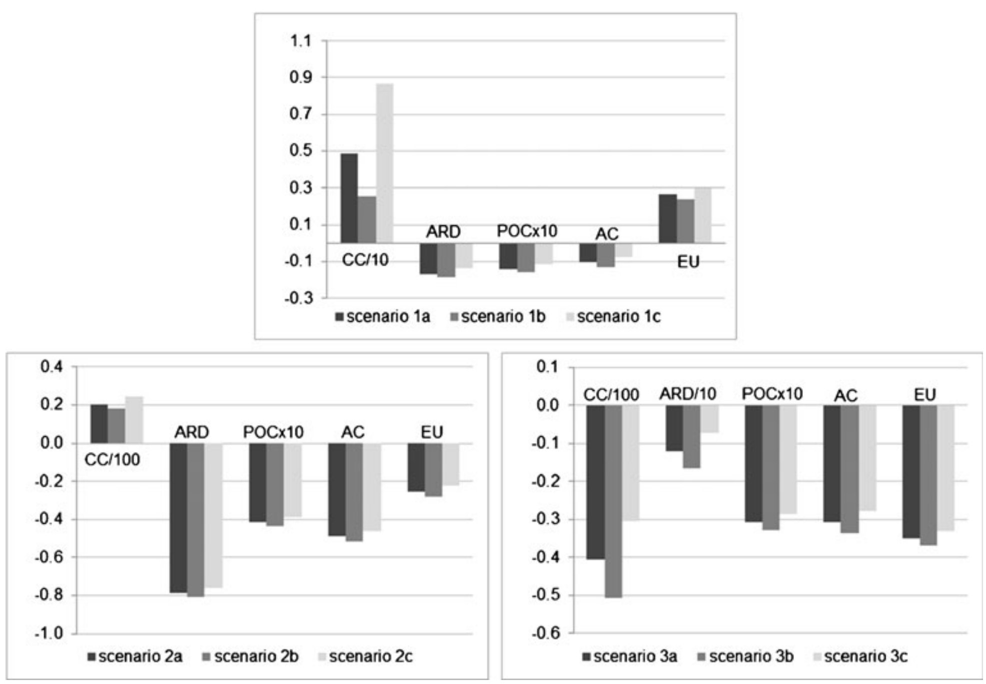


Fig. 5 Sensitivity analysis results for scenarios 1, 2, and 3 using parameters with higher uncertainty

Table 6 Recycling and recovery rates achieved in each scenario

Scenarios	RATE [%]	
	Recycling	Recovery
Scenario 1	84.2	88.3
Scenario 2	84.2	95.5
Scenario 3	84.6	96.1
Targets of European Union policy (Directive 2000/53/EC) for the year 2015	85.0	95.0

advantages resulting from the opportunity to operate in co-incineration with MSW, ASR co-incineration should not be considered as a long-term alternative to landfill since it does not allow the accomplishment of 85 % recycling target set by the European Union policies. This scenario has a significant impact on the climate change potential due to emissions from the thermal oxidation of polymeric materials present in the ASR fraction.

3. Scenario 3 seems to ensure a net environmental upgrading, which includes the supplementary dismantling of components for recycling and SRF production and its burning in the cement industry.

The proposed additional dismantling of ELV not only contributes to environmental benefits but also meets the European recovery and recycling targets for this waste stream. However, to achieve the Directive's requirements, an increase in labor is required. Additional recycling material and energy recovery revenues can compensate the increase of dismantling costs. A social benefit results in an increase of employment.

The dismantlers should be encouraged to carry out this type of operation. This may be possible with the creation of recycling companies, awards for the use of recycled materials, the imposition of surcharges on virgin raw materials or payments to recyclers in order to increase the market value of secondary materials. A quota of vehicle license tax or a fee paid when purchasing a new vehicle may be used to pay for the system.

In order to attain an environmentally sound treatment of ELV, it would be helpful, in addition to the recovery and recycling targets, to establish treatment obligations for particular material streams, taking into account their overall environmental impacts. Since currently in Portugal 60 vehicle dismantlers (Valorcar 2011) and all vehicle recycling factories operate independently, the authors suggest that greater recycling efficiencies could be achieved if these two operations were combined.

New devices/tools that allow a greater efficiency in the dismantling process should be developed. On the other hand, PST for physical and mechanical upgrading of ASR should be continuously researched for its application in shredding companies.

Industrial applications by shredding companies, using these PSTs (e.g., Galloo, VW-SiCon, Argonne, MBA-polymers, Salyp process, Stena and R-plus), have shown that the European targets can indeed be reached, showing that these approaches are promising and sustainable solutions for the future ELV treatment (GHK/Bios 2006; Reinhardt 2005; Ferrão et al. 2006; Nourreddine 2007; Jody and Daniels 2006; Krinke et al. 2006). Additionally, the end-of-life treatments should be integrated into eco-design strategies aiming at more efficient separation of high added value materials such as metals and plastics.

Environmental policies differ between European Member States, but there is a strong belief that increases in landfill restrictions represents one of the best drivers to get eco-efficient treatments.

To conclude, it is important to state the limitations in this LCA study:

- LCA can be resource and time intensive. Gathering the data can be problematic, and the availability of data can greatly influence the accuracy of the results.
- Considering only the ecological criteria does not include economic and social aspects;
- Results of LCA studies focused on global or regional issues may not be appropriate for local applications, i.e., local conditions might not be adequately represented by regional or global conditions.
- It is a methodology that is constantly evolving and, as such, different approaches to the problem can result in different outcomes;
- The lack of spatial and temporal dimensions in the inventory data used for impact assessment introduces uncertainty in impact results. This uncertainty varies with the spatial and temporal characteristics of each impact category.

Despite these limitations, the authors recommend the use of their results in order to identify the main opportunities for improving the environmental performance of ELV life cycle management.

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References

Al-Salem SM, Lettieri P, Baeyens J (2010) The valorization of plastic solid waste (PSW) by primary to quaternary routes: from re-use to energy and chemicals. Prog Energ Combust 36(1):103–129

CEN (2006) CEN/TS 15359:2006, solid recovered fuels—specifications and classes

Ciacci L, Morselli L, Passarini F, Santini A, Vassura I (2010) A comparison among different automotive shredder residue treatment processes. Int J Life Cycle Assess 15(9):896–906

Classen M, Althaus HJ, Blaser S, Tuchschmid M, Jungbluth N, Doka G, Faist Emmenegger M, Scharnhorst W (2009) Life cycle inventories of metals, final report ecoinvent data v2.1, No 10, EMPA Dübendorf, Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland

Dias S (2011) Produção de CDR certificado e Mercado. Instituto Superior Técnico, Lisboa

Ecoinvent (2010) Ecoinvent v2.2, Swiss Centre for Life Cycle Inventories. http://www.ecoinvent.org/. Accessed March 2011

EDP (2011) Origem da eletricidade fornecida em 2011. https://energia.edp.pt/apoio-ao-cliente/origem-energia.aspx. Accessed June 2011

ELCD (2010) European Reference Life Cycle Database. http://lct.jrc.ec.europa.eu/assessment/tools. Accessed May 2011

EU (2000) Directive 2000/53EC of the European Parliament and of the Council (2000.9.18)

Eurostat (2012) End of life vehicles (ELVs). http://epp.eurostat.ec.europa.eu/portal/page/portal/waste/key_waste_streams/end_of_life_vehicles_elvs. Accessed September 2012

Ferrão P, Amaral J (2006) Assessing the economics of auto recycling activities in relation to European Union Directive on end of life vehicles. Technol Forecast Soc 73(3):277–289

Ferrão P, Nazareth P, Amaral J (2006) Strategies for meeting EU end-of-life vehicle reuse/recovery targets. J Ind Ecol 10(4):77–93

Fink JK (1999) Pyrolysis and combustion of polymer wastes in combination with metallurgical processes and the cement industry. J Anal Appl Pyrolysis 51(1–2):239–252

GHK/Bios (2006) A study to examine the benefits of the End of Life Vehicles Directive and the cost and benefits of a revision of the 2015 targets for recycling, re-use and recovery under the ELV Directive, Final Report to DG Environment, Birmingham

Heijungs R, Guinée JB, Huppes G, Lankreijer RM, Sleeswijk A (1992) Environmental life cycle assessment of products—guide and backgrounds. C. o. E. Science, Leiden University, Leiden

IPCC (2006) IPCC Guidelines for National Greenhouse Gas Inventories. National Greenhouse Inventories Programme

ISO (2006a) Environmental management—life cycle assessment—principles and framework. ISO 14040, International Organization for Standardization, Geneva, Switzerland

ISO (2006b) Environmental management—life cycle assessment—requirements and guidelines. ISO 14044, International Organization for Standardization, Geneva, Switzerland

Jeong KM, Hong SJ, Lee JY, Hur T (2007) Life cycle assessment on end-of-life vehicle treatment system in Korea. J Ind Eng Chem 13(4):624–630

Jody BJ, Daniels EJ (2006) End-of-life vehicle recycling: the state of the art of resource recovery from shredder residue. Energy Systems Division, Argonne National Laboratory, Lemont

Kellenberger D, Althaus HJ, Jungbluth N, Künniger T, Lehmann M, Thalmann P (2007) Life cycle inventories of building products. Final report ecoinvent Data v2.0 No. 7, EMPA Dübendorf, Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland

Krinke S, Boßdorf-Zimmer B, Goldmann D (2006) The Volkswagen-SiCon Process: eco-efficient solution for future end-of-life vehicle treatment. In: LCE2006, 13th CIRP International Conference on Life Cycle Engineering, Belgium

Ladeira S (2002) Modelo Técnico-económico da infra -estrutura nacional de processamento de veículos em fim de vida útil. Instituto Superior Técnico, Lisboa

Le Borgne R, Feillard P (2001) End-of-life of a polypropylene bumper skin. Int J Life Cycle Assess 6(3):167–176

Morselli L, Santini A, Passarini F, Vassura I (2010) Automotive shredder residue (ASR) characterization for a valuable management. Waste Manage 30(11):2228–2234

Nourreddine M (2007) Recycling of auto shredder residue. J Hazard Mater 139(3):481–490

NP (2008) NP 4486:2008, Combustíveis derivados de resíduos—Enquadramento para a produção, classificação e gestão da qualidade

Puri P, Compston P, Pantano V (2009) Life cycle assessment of Australian automotive door skins. Int J Life Cycle Assess 14(5):420–428

Redin LA, Hjelt M, Marklund S (2001) Co-combustion of shredder residues and municipal solid waste in a Swedish municipal solid waste incinerator. Waste Manag Res 19(6):518–525

Reinhardt WA (2005) Drive towards compliance—recycling end-of-life vehicles in an enlarged EU. Waste Management World 4:53–62

Roy C, Chaala A (2001) Vacuum pyrolisis of automobile shredder residues. Resour Conserv Recycl 32:1–27

Santini A, Herrmann C, Passarini F, Vassura I, Luger T, Morselli L (2010) Assessment of ecodesign potential in reaching new recycling targets. Resour Conserv Recycl 54(12):1128–1134

Sawyer-Beaulieu S, Tam E (2005) Applying life cycle assessment (LCA) to North American end-of-life vehicle (ELV) management processes. SAE Technical Paper Series, SAE International 2005-01-0846

Sawyer-Beaulieu S, Tam E (2008) Constructing gate to gate lie cycle inventory (LCI) of end-of-life vehicle (ELV) dismantling and shredding processes. SAE Technical Paper Series, SAE International 2008-01-1283

Schmidt W-P, Dahlqvist E, Finkbeiner M, Krinke S, Lazzari S, Oschmann D, Pichon S, Thiel C (2004) Life cycle assessment of lightweight and end-of-life scenarios for generic compact class passenger vehicles. Int J Life Cycle Assess 9(6):405–416

Valorcar (2011) Relatório de Atividade 2010. http://www.Valorcar.pt/docs/REL_ACT_2010_VFV_FINAL.pdf. Accessed February 2011

Zoboli R, Barbiroli G, Leoncini R, Mazzanti M, Montresor S (2000) Regulation and innovation in the area of end-of-life vehicles. IDSE-CNR, Milan